Laboratory Sediment Bioassay Report on Stormwater Pond Sediments in Guelph and Greater Toronto Area, Ontario 1997

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Prepared by:
D. Bedard
Ontario Ministry of the Environment
Standards Development Branch
125 Resources Road
Etobicoke, Ontario

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EXECUTIVE SUMMARY

Sediments were collected from 15 stormwater ponds in July and September 1997, as part of an assessment to characterize the potential ecological impacts of sediments associated with these man-made structures. Nine stormwater ponds were sampled in Guelph, Ontario in primarily residential areas. Six stormwater ponds were sampled within the Greater Toronto Area (GTA) and depending on the site had industrial, commercial or residential land usage. The median age for the Guelph and GTA ponds were 13 years and 8 years, respectively.

An objective of this study was to assess the spatial pattern of sediment toxicity and chemical bioaccumulation using static, laboratory sediment toxicity tests. Three independent toxicity tests were performed on whole-sediment samples. Mortality and growth of the burrowing mayfly, *Hexagenia limbata*, was measured in 21-day exposures. Chironomid (*Chironomus tentans*) growth and mortality was determined in 10-day tests. Mortality and chemical uptake by the juvenile fathead minnow, *Pimephales promelas*, was examined by a standard 21-day test.

Mayfly survival, mayfly growth and midge growth were significantly reduced at one of the GTA test sites (station T-3). Mayfly survival was 74% and benthic growth reduction was at least 50% relative to the weights obtained by the control animals or relative to all other GTA test sites. Only one of the Guelph sites (station G-8) resulted in slight growth impairment based on one of the five test endpoints. In total, 13 of 15 of the test sediments were categorized as being nontoxic. Spearman rank correlation analysis indicated sediment oils and greases concentrations in the test sediments was the best predictor of biological effects. A threshold concentration of 400 µg/g (corrected for organic carbon) was obtained for describing the likelihood of a lethal effect and 200 µg/g OC with regard to a sublethal effect. This relationship applied to both the Guelph and GTA studies, albeit on a small number of toxic samples, and may potentially serve as a marker of sediment toxicity.

A site-specific characteristic that applied to all of the Guelph stormwater pond sediments was elevated un-ionized ammonia present during the fathead minnow bioassay. On average, concentrations in the Guelph exposures were three times higher than those found in the corresponding minnow test conducted on the GTA sediments and reached chronic-effect level concentrations for fathead minnows.

In summary, most of the Guelph and GTA stormwater pond test sediments were of a high sediment quality or non-impacted based on a lack of significant adverse biological effects. Multiple biological effects were noted at one of the GTA sites, situated in a commercial / light industrial area. Whole-organism organic chemical concentrations measured on laboratory-exposed fathead minnows were often comparable with those found in the control fish and did not exceed federal or provincial tissue guidelines. The exact extent to which chemicals in stormwater pond sediments can bioaccumulate in the aquatic food chain is best measured in the field on a larger number of samples.

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1.0 INTRODUCTION

In 1997, Environment Canada (EC), Environmental Conservation Branch (ECB) and the Ontario Ministry of the Environment (OMOE), Standards Development Branch (SDB), undertook a study to assess sediment quality in a number of stormwater ponds in the cities of Guelph and the Greater Toronto Area (GTA), Ontario. The work was funded through the Great Lakes 2000 Clean-up Fund. The study objectives included (1) an assessment on the degree to which vertebrates and invertebrates use stormwater ponds; (2) the extent of inorganic and organic contamination in water, sediment and biota from urban stormwater ponds; and (3) to evaluate the toxicity of stormwater pond water and sediment on aquatic life.

Numerous reports have been written that describe the physical attributes of sediments found in stormwater ponds and to a lesser extent, the degree of chemical contamination (see Wren *et al.*, 1997; Watt and Marsalek; 1994; Marsalek and Marsalek, 1997). Stormwater ponds are designed to capture suspended particulates found in stormwater runoff and reduce the release of harmful pollutants into local water courses. The sediment that is deposited may act as reservoirs of inorganic and organic contaminants within the system. Substantial sediment accumulation rates have been measured at 1 to 4 cm per year (Yousef *et al.*, 1994). Eventually the loading capacity of a stormwater pond is attained and the question of sediment removal and disposal becomes an issue (Marsalek and Larkin, 1998).

An area of increasing concern is the degree of risk associated with sediments to both resident species and those species that frequent these areas. Direct and indirect exposure to bottom sediments could lead to the impairment of benthic organisms and chemical biomagnification. Biomagnification is a term that describes the transfer of a chemical from food to consumer, such that the residue concentration increases from one trophic level to the next (Rand and Petrocelli, 1985). This process pertains to both aquatic and terrestrial food chains, as well as their interaction.

Laboratory sediment toxicity tests are a useful tool for examining biological effects and chemical uptake and availability and is part of an integrated approach in evaluating sediment quality (Jaagumagi and Persaud, 1996). Only recently has a greater emphasis been placed on examining the toxicological significance of contaminants associated with stormwater pond sediments (Dutka *et al.*, 1994; Wenholz and Crunkilton, 1995; Karouna-Renier and Sparling, 1997; Steevens *et al.*, 1998). Many of these studies employed non-benthic test species and manipulated sediment samples e.g. interstitial water, which may not necessarily maximize contaminant routes of exposure and availability, or be less ecologically relevant (Harkey *et al.*, 1994). Other studies have documented the biological impact of other stormwater-related events such as road runoff (Maltby *et al.*, 1995; Boxall and Maltby, 1997).

Historical data bases of water and sediment quality are available for several of the Guelph and GTA stormwater pond sites, which made them good candidates for further detailed investigations as described by Dunn (1997). Field surveys were conducted in order to measure the variety and abundance of wildlife at each site using standard protocols (Wren *et al.*, 1997). In-place monitoring of water quality and quantity to assess performance efficiency continues at some sites through the Stormwater Assessment and Monitoring Performance Program (SWAMP). An attempt was made to select ponds of various ages in order to examine relative differences in

chemical concentrations and associated biological effect.

This report provides the results and interpretation of whole-sediment toxicity tests conducted by OMOE, SDB following documented laboratory sediment toxicity test methods (Bedard *et al.*, 1992). Two separate series of toxicity tests were completed in 1997. Nine stormwater ponds or SWPs from Guelph were tested in the spring, followed by six GTA SWPs in the fall. Each sediment was tested using the mayfly nymph, *Hexagenia limbata* (21-day exposure, survival and growth), the midge larvae, *Chironomus tentans* (10-day exposure, survival and growth) and the juvenile fathead minnow, *Pimephales promelas* (21-day exposure, survival and chemical bioaccumulation). The battery of sediment toxicity tests provides several endpoints using organisms representing different trophic levels in order to measure differences in sediment quality. The laboratory toxicity tests are a cost-effective technique for determining whether sediment-associated chemicals are harmful to benthic organisms or are being released into the water column. In conjunction with an appropriate negative control sediment, spatial differences in sediment quality, the relative availability of contaminants and their potential impacts can be ascertained. Most of the sediment contaminant concentrations were based on samples prepared for laboratory toxicity testing. The sediment was analysed for particle size, nutrients, metals, total PCBs, organochlorine pesticides and chlorinated benzenes. Oils and greases concentrations were determined on field samples. Surviving fathead minnows were submitted for whole-body tissue analysis of total PCBs, chlorinated organics and pesticides.

2.0 MATERIALS AND METHODS

2.1 Sample Collection and Site Description

In mid-July 1997, surficial sediment was collected at nine locations in the city of Guelph, Ontario (Table 1). Test sediment was collected in late September 1997 at six study locations for the GTA SWP survey (Table 2). The sampling locations were designated by EC, Environmental Conservation Branch and Toronto Region Conservation Authority, for water and sediment chemistry, field benthic community analysis, laboratory sediment toxicity testing and wildlife survey (Dunn,1997).

Sampling was done using a mini-Ponar grab sampler. At each station, approximately 15 L of composited surficial sediment (top 10 cm) was collected from several grabs along a transect, in order to obtain a representative sample. The composited sediment was placed into 20 L plastic buckets lined with food-grade polyethylene bags and transported to the Toronto, Ontario laboratory where they were stored at 4°C until required.

In the Guelph study, the urban SWPs ranged in age from 9 years (station G-1) to 22 years (station G-4) with a median age of 13 years and each were located in residential areas. Among the GTA sites, the youngest SWP was station T-1; 3 years, and the oldest was 10 years (stations T-4 and T-5). The median age was 8 years. Land use varied from residential, commercial and light industrial, depending on location.

TABLE 1. 1997 Guelph Stormwater Pond Age and Land Use

Station Number	Pond Age (years)	Land Use
G - 1	9	Residential
G - 3	19	Residential
G - 4	22	Residential
G - 5	16	Residential
G - 6	13	Residential
G - 7	11	Residential
G - 8	15	Residential
G - 9	13	Residential
G - 20	~10	Residential

TABLE 2. 1997 GTA Stormwater Pond Age and Land Use

Station Number	Pond Age (years)	Land Use
T - 1	3	Commercial
T - 2	6	Residential
T - 3	7	Commercial / Light Industrial
T - 4	10	Residential
T - 5	10	Residential
T - 6	6	Residential / Commercial

Neither study included a reference control site which would have been indicative of local background contaminant conditions. Normally sediment toxicity tests include a reference control sediment that is used as a measure biological effects due to sediment type and low-level sediment concentrations. It was considered that, at the time the samples were collected, no suitable reference areas were available. This was in part due to the nature of the sites being studied e.g. man-made structures. Instead, emphasis will be placed on any among site differences in biological response. Sediment collected from Honey Harbour, Georgian Bay, Ontario served as the negative control sediment for both bioassays. The negative control sediment is a relatively uncontaminated sediment that provides a measure of test acceptability (ASTM, 1997a).

2.2 Analytical Methods

Chemical analysis of sediment samples was conducted by the OMOE, Laboratory Services Branch, located in Toronto. Test methods are described in the *OMOE Handbook of Analytical Methods for Environmental Samples* (OMOE, 1983). Chemical analysis of fish samples was completed by the National Wildlife Research Center, Laboratory Services Section, located in Hull, Quebec. Quality assurance procedures included method blanks, spikes, duplicates and standard reference materials.

Sediment Nutrients and Particle Size Characterization

Homogenized bulk sediment (<2 mm fraction) was measured for total phosphorus (TP), total Kjeldahl nitrogen (TKN) and percent weight loss on ignition (LOI) which measured approximate organic content. Sediment total organic carbon (TOC) was determined with a LECO carbon analyzer using a dry combustion technique which oxidized carbon to CO_2. Particle size was measured on 50 g, air-dried samples using a Microtrac particle size analyzer for the size range 1.0 mm to 0.1 µm. This gives data on percent sand (2mm -62 µm), percent silt (62- 3.7 µm) and percent clay (3.7 - 0.1 µm) size classes. Detailed test methodology is described in OMOEE (1995a; 1995b).

Trace Metals in Sediment

Prepared sediment samples were digested using a concentrated aqua-regia acid mixture (1 part HNO_3 to 3 parts HCl). The dissolved trace metals including As, Cd, Cr, Cu, Fe, Pb, Mn, Ni and Zn in the digestates were detected by inductively coupled argon plasma atomic emission spectroscopy (ICP-AES), and Hg by flow injection vapour generated flameless atomic absorption spectroscopy (AAS). Detailed test methodology is described in OMOEE (1994a).

Organic Chemicals in Sediment

Moist sediment samples were solvent-extracted with acetone and dichloromethane. The extract was transferred to a rotary evaporator, concentrated and fractionated on a Florisil column. Different solvent combinations were used to elute the extracts into three groups: fraction A1

contained total PCBs, five Aroclor groups, hexachlorobenzene, heptachlor, aldrin, octachlorostyrene, pp-DDE and mirex; fraction A2 contained a- & b-BHC, a- & b-chlordane, op-DDT, pp-DDD, pp-DDT; and fraction A3 included heptachlor epoxide, oxychlordane, dieldrin, endosulfan I & II, endosulfan sulphate, endrin and methoxyclor. Analytes were identified and quantified using capillary gas chromatography equipped with a Ni^{63} electron capture detector (GLC-ECD). The detection limit for total PCBs is 20 ng/g dw. Detailed test methodology is described in OMOEE (1994b).

Organic Chemicals and Percent Lipid in Biota

Pooled whole fish samples (~5 g) were thawed, homogenized and extracted on a single sample per station in both studies. The homogenate was dehydrated with anhydrous sodium sulfate and extracted with dichloromethane (DCM) : hexane (1:1 v/v). The extract was evaporated and then extracted on a deactivated Florisil column. Quantitative analysis was performed using capillary gas chromatography equipped with a mass selective detector. Final results are reported on a wet weight basis for total PCBs and DDT compounds. The limit of detection for total PCBs is 5 ng/g ww and 2.5 ng/g ww for organochlorine pesticides. Percent lipid was also determined on a subsample. Analytical methods are similar to those described by Peakall *et al.*, (1986) and Turle *et al.*, (1991).

2.3 Laboratory Biological Testing Methods

Basic Experimental Design

Sediment biological tests were conducted according to OMOE standardized procedures (Bedard *et al.*, 1992) and are briefly described below. The bioassays were static, single-species tests using whole-sediment. The experimental unit was a 1.8 L test chamber containing prepared sediment and dechlorinated municipal tap water (1:4, v:v). The chambers were randomly placed into a holding tank at ambient room temperature and maintained under a 16:8 hour, light:dark photoperiod and continuous aeration.

Moist field-collected bottom sediment was pressed through a 2-mm stainless-steel sieve to remove existing large biota and debris prior to use. Sieving was completed from July 18 - 22, 1997 for the Guelph study and from September 29 - October 1, 1997 for the GTA study. Subsamples of this homogenized sediment were submitted for chemical and physical characterization according to standard OMOE procedures (OMOE, 1989). The sieved sediment was homogenized with a spatula and stored in 4 L acid-rinsed glass jars until required. Three hundred and twenty-five millilitre aliquots of homogenized sediment were placed into the test chamber and overlaid with the test water. After settling overnight, the chambers were aerated continuously until the end of the test. A clean, negative control sediment was collected from Honey Harbour, Georgian Bay. Negative control mortality must not exceed 15% for mayflies and fathead minnows and 25% for chironomids or the test is declared invalid.

Water in the exposure chambers was regularly monitored for pH, conductivity, total ammonia, un-ionized ammonia and dissolved oxygen. Dead organisms were removed and the

numbers recorded daily in the mayfly and minnow tests. Any signs of abnormal behaviour of the test organisms or changes in appearance of the test chambers were noted. Water loss due to evaporation was replenished as needed.

Hexagenia limbata Lethality and Growth Assay

The Guelph toxicity test used four month old laboratory-reared mayfly nymphs with an average wet weight of 4.22 mg ± 0.29 (s.e.) (n=37). The GTA toxicity test used five month old nymphs that weighed 5.14 mg ± 0.45 (s.e.) (n=35). The nymphs were raised from field-collected eggs obtained by Dr. J. Ciborowski at the University of Windsor, Windsor, Ontario. Mayflies were reared according to OMOE procedures (Bedard *et al.*, 1992) and methods described by Friesen (1981).

The rearing procedure involved the transfer of 600 newly-hatched nymphs to a 6.5 L aquarium which contained 2 cm of autoclaved sediment and 5.6 L dechlorinated tap water. Animals were maintained at ambient room temperature, 16:8 hour, light:dark photoperiod, continuous aeration and fed a vegetable diet.

Test organisms were retrieved from the rearing aquaria by sieving small portions of sediment in a 500-µm mesh brass sieve. The nymphs were washed into an enamel tray which held a fine mesh sieve and aerated, dechlorinated water. A Pasteur pipette (5-mm opening) was used to transfer the mayflies into 100 mL beakers of water and the contents were gently poured into the test chambers. Each test involved adding ten nymphs to each of the three replicate test chambers for a period of 21 days. Animals were not fed during the length of the test.

At the end of the test, the contents of each test chamber were emptied and rinsed in a sieve bucket. Surviving animals were counted and transferred to 150 mL beakers holding 100 mL dechlorinated water. The nymphs were immobilized with Alka-Seltzer®, blotted dry and individuals weighed to the nearest 0.01 mg.

Chironomus tentans Lethality and Growth Assay

Each toxicity test used 10-12 day old, cultured chironomid larvae weighing an average wet weight of less than 1 mg. The OMOE maintains continuous cultures of *C. tentans* larvae from egg to adult, following standard methods (Bedard *et al.*, 1992, Mosher *et al.*, 1982, Townsend *et al.*, 1981). Egg masses were originally supplied by Dr. J. Giesy at Michigan State University, Lansing, Michigan and have been cultured for several generations in our laboratory.

Initially, the midges were reared in enamel trays for 10 to 12 days and then maintained in a 21 L aquarium containing 1.6 L of silica sand. The cultures were held at ambient room temperature with continuous aeration and under a 16:8 hour, light:dark photoperiod. The larvae were provided a vegetable diet *ad libitum*.

Second and third instar larvae were directly transferred from the enamel rearing trays into the test chamber using the 5-mm opening of a Pasteur pipette. A total of 15 animals were added

per chamber to each of the three replicates for both studies. Animals were fed daily 30 mg of a Cerophyll®:Tetra Conditioning Vegetable® (3:2, w:w) diet.

After 10 days (GTA study) or 11 days (Guelph study), the contents of the test chambers were emptied and washed in a sieve bucket. The latter test duration of 11 days is a slight deviation from the standard test duration of 10 days and represents $1/10^{th}$ the test duration. This deviation is not expected to contribute significantly to the test outcome. Surviving animals were sorted, removed and placed into 150 mL beakers holding 100 mL dechlorinated water and 15 mL silica sand. The larvae were counted, blotted dry and individuals weighed to the nearest 0.01 mg.

Pimephales promelas Lethality and Bioaccumulation Assay

The tests used cultured, juvenile fathead minnows with an average wet weight of 409 mg ± 18 (s.e.) (n=44) and 430 mg, for the Guelph and GTA toxicity tests, respectively. The latter value was an estimate of the starting animal size based on a reduced subsample of twelve animals. The minnows were cultured at the OMOE laboratory following techniques which for the most part are US EPA procedures (USEPA, 1987) with minor revisions (Bedard *et al.*, 1992).

Cultures were maintained at 20°C in a flow-through dechlorinated water system and under a 16:8 hour, light:dark photoperiod. Breeders were kept in 60 L glass aquaria and eggs were laid on spawning tiles. The tiles were incubated in a 25°C water-bath and the developing larvae were transferred to 400 L fibreglass holding tanks. Larval fish were fed 48-hour old live brine shrimp while juveniles and breeders were provided frozen brine shrimp. Each size class was fed *ad libitum*.

Each test chamber received 10 juvenile minnows for triplicate jars for the Guelph study and 8 minnows for triplicate jars for the GTA study. The minnows were sorted into 250 mL glass beakers in groups of four or five. The contents of the beakers were emptied into a small net and the minnows released into the test chamber.

The minnows were exposed for 21 days and fed a Tetra Conditioning Vegetable® diet daily in an amount equivalent to 1% of the average starting wet weight. After 21 days the surviving fathead minnows were pooled from each replicate, counted, immobilized with Alka-Seltzer® and placed into 30 mL glass vials and frozen pending chemical analysis.

Reference Toxicant Testing

Water-only reference toxicity ($CuSO_4$) tests were conducted with *H. limbata* and *C. tentans* for 48-hours and LC50s were calculated for each study. The static tests consisted of four test concentrations and a control. The nominal copper concentrations were 0.05, 0.25, 0.5, 1.0 and 3.0 mg/L. Ten mayfly nymphs or midge larvae were placed into each of four replicate 250 mL beakers. To help reduce stress, five glass tubes were placed into the mayfly test beakers and a fine layer of silica sand was added to the midge test containers. Water quality parameters were recorded at 0 and 48 hours.

The mayfly test used four month old laboratory-reared mayfly nymphs with an average wet weight of 4.01 mg ± 0.25 (s.e.) for the Guelph study; and 5.64 mg ± 0.51 (s.e.) for the GTA study. The midge larvae were 10-12 day post-hatch with an average wet weight < 1 mg in each set of tests.

Bioassay Schedule for Guelph stormwater pond 1997 Sediment Samples

Test Organism	Species	Starting Date ('97)	Completion Date ('97)	Test Duration
Mayfly	*Hexagenia limbata*	July 29	August 19	21 days
Chironomid	*Chironomus tentans*	July 25	August 5	11 days
Minnow	*Pimephales promelas*	July 25	August 15	21 days

Bioassay Schedule for GTA stormwater pond 1997 Sediment Samples

Test Organism	Species	Starting Date ('97)	Completion Date ('97)	Test Duration
Mayfly	*Hexagenia limbata*	October 17	November 7	21 days
Chironomid	*Chironomus tentans*	October 10	October 20	10 days
Minnow	*Pimephales promelas*	November 5	November 26	21 days

2.4 Statistical Methods

Statistical analyses were performed using the SAS® software package (SAS, 1985). Comparisons were made among the test and control sediments using One-Way Analysis of Variance (ANOVA) and Tukey's studentized range test (HSD) and planned comparisons (Steel and Torrie, 1960). Dunnett's one-tailed *t*-test was used solely to compare mortality between the control and test sediments and the associated minimum significant difference (MSD) was described as a measure of test sensitivity. Analysis was made on arc-sine transformed mortality data. Homogeneity of variance across groups was tested using Bartlett's test. Coefficients of variation (C.V. %) were calculated for each endpoint as a measure of test precision. Spearman rank correlation analysis was used to investigate the correlation among the different biological endpoints for each species and sediment characteristics. LC50's (including the associated 95% confidence

limits) were calculated using software developed by Stephan (1977) and were estimated by probit analysis.

3.0 RESULTS

3.1 Water Quality Test Parameters

Conductivity, pH, total ammonia, un-ionized ammonia and dissolved oxygen parameters were periodically measured on the overlying water for each test species in each of the two sets of tests (Tables 3 and 4). Values are reported as mean ± standard deviation.

Similar pH water quality measurements were recorded among the test sites, regardless of test species or study location. The average pH recorded among the test sediments for the Guelph and GTA SWPs were 8.0 and 8.2, respectively, with values ranging from 7.6 to 8.4. Generally, conductivity readings were similar among sites and between tests. The average conductivity among the nine Guelph test sediments for the mayfly, midge and minnow tests were 566 µmho/cm, 529 µmho/cm and 642 µmho/cm, respectively. Among the GTA sites, station T-2 consistently had conductivity readings about twice the value reported for the remaining test sediments. For example, conductivity for station T-2 exposures ranged from 946 µmho/cm to 1134 µmho/cm among the three test species, as compared to the average conductivity of 558 µmho/cm to 668 µmho/cm calculated for the other five test sediments. Dissolved oxygen within the test jars remained above the minimum acceptable level (>4 mg/l) throughout the test (OMOEE, 1994c). Test temperature was at or near 20°C for each bioassay.

The amount of total ammonia (NH_4), along with the converted un-ionized ammonia (NH_3) based on temperature and pH, were also recorded (Tables 3 and 4). Both set of tests showed similar trends in ammonia concentrations. Average concentrations were lowest in the mayfly test (0.02 and 0.03 mgNH_3/L) and highest in the minnow test (0.08 and 0.34 mgNH_3/L). The midge test had intermediate concentrations of 0.04 and 0.06 mgNH_3/L. In addition, the number of exceedences above the PWQO of 0.02 mgNH_3/L was fewest for the mayfly exposures and relatively equal in both the midge and minnow bioassays. None of the un-ionized ammonia concentrations were above cited acute concentrations for fathead minnows (Thurston *et al.*,1983) and midge larvae (Schubauer-Berigan *et al.*, 1995; Whiteman *et al.*, 1996). However, each of the Guelph SWP sediments resulted in values that approached or surpassed the chronic-effect *Pimephales promelas* threshold concentration of 0.27 mgNH_3/L, for juvenile fathead minnows of a similar size (Thurston *et al.*, 1986).

3.2 Sediment Characterization

The following sections summarize the sediment physical and chemical parameters to aid in the interpretation of the biological toxicity results. Chemical analysis is based on the sediment prepared for toxicity testing and results may differ from those reported for any field samples collected concurrently. Any dissimilarities are likely due to *in-situ* chemical heterogeneity and/or sampling depth and sample handling (ASTM, 1997b).

TABLE 3. Mean (± s.d.) water quality characteristics in Guelph SWP 1997 sediment bioassays.

Test Organism: Mayfly (Hexagenia limbata)[a]			Test Temperature: 20.3°C (0.5)		
Station	*pH*	*D.O.* mg/L	*Conductivity* umho/cm	*Total Ammonia* mg/L	*Un-ionized Ammonia* mg/L
Control	7.78 (.06)	8.7 (.01)	302 (6)	<0.10 (0)	<0.003
Stn G - 1	8.07 (.13)	8.5 (0.2)	471 (67)	0.58 (0.56)	0.022
Stn G - 3	8.14 (.18)	8.4 (0.2)	699 (126)	0.19 (0.19)	0.009
Stn G - 4	8.19 (.09)	8.5 (0.2)	530 (90)	0.65 (0.85)	0.024
Stn G - 5	8.11 (.04)	8.5 (0.2)	505 (103)	0.34 (0.44)	0.021
Stn G - 6	8.18 (.03)	8.4 (0.3)	760 (144)	0.72 (0.80)	0.028
Stn G - 7	8.18 (.08)	8.5 (0.2)	518 (76)	0.40 (0.60)	0.016
Stn G - 8	8.09 (.08)	8.5 (0.2)	476 (78)	0.42 (0.65)	0.016
Stn G - 9	8.19 (.04)	8.5 (0.2)	537 (75)	1.95 (2.13)	0.074
Stn G - 20	8.16 (.18)	8.5 (0.1)	599 (91)	0.56 (0.46)	0.027
Test Organism: Midge (Chironomus tentans)[b]			Test Temperature: 19.8°C (0.8)		
Station	*pH*	*D.O.* mg/L	*Conductivity* umho/cm	*Total Ammonia* mg/L	*Un-ionized Ammonia* mg/L
Control	7.78 (.02)	8.8 (0.1)	309 (5)	0.15 (0.08)	<0.003
Stn G - 1	8.13 (.08)	8.6 (0.2)	453 (62)	1.16 (1.22)	0.045
Stn G - 3	8.12 (.09)	8.6 (0.1)	640 (87)	0.58 (0.62)	0.023
Stn G - 4	8.20 (.05)	8.7 (0.1)	498 (63)	1.83 (1.60)	0.073
Stn G - 5	8.10 (.11)	8.5 (0.3)	442 (59)	0.96 (0.96)	0.037
Stn G - 6	8.32 (.10)	8.6 (0.3)	680 (91)	1.50 (0.70)	0.127
Stn G - 7	7.98 (.39)	7.2 (2.6)	507 (59)	1.46 (1.40)	0.031
Stn G - 8	8.09 (.07)	8.6 (0.3)	452 (62)	0.93 (0.90)	0.035
Stn G - 9	8.07 (.15)	8.2 (0.6)	505 (54)	2.99 (2.76)	0.113
Stn G - 20	8.13 (.17)	8.5 (0.4)	590 (102)	1.51 (1.21)	0.065

[a] Sample size N=4; [b] Sample size N=3;

Underlining indicates un-ionized ammonia concentrations that exceed the PWQO of 0.02 mg/L

TABLE 3. Continued.

Test Organism: Minnow (Pimephales promelas)[a]			Test Temperature: 20.0°C (0.7)		
Station	*pH*	*D.O.* mg/L	*Conductivity* umho/cm	*Total Ammonia* mg/L	*Un-ionized Ammonia* mg/L
Control	7.15 (.49)	7.9 (1.0)	332 (30)	1.36 (1.44)	0.009
Stn G - 1	7.90 (.14)	8.3 (0.4)	519 (87)	7.42 (8.28)	0.281
Stn G - 3	7.65 (.20)	7.6 (1.0)	813 (197)	8.57 (6.81)	0.225
Stn G - 4	7.88 (.30)	8.1 (0.5)	595 (118)	10.45 (12.15)	0.396
Stn G - 5	7.80 (.17)	8.1 (0.6)	559 (136)	10.05 (6.35)	0.277
Stn G - 6	7.74 (.21)	6.9 (1.6)	835 (169)	15.60 (13.38)	0.397
Stn G - 7	7.95 (.11)	8.3 (0.1)	618 (131)	8.65 (9.84)	0.328
Stn G - 8	7.71 (.11)	8.0 (0.4)	597 (148)	15.00 (11.01)	0.342
Stn G - 9	7.93 (.19)	8.0 (0.3)	596 (108)	16.72 (9.68)	0.564
Stn G - 20	7.73 (.10)	7.9 (0.1)	653 (114)	11.35 (7.69)	0.293

[a] Sample size N=4.

Underlining indicates un-ionized ammonia concentrations that exceed the PWQO of 0.02 mg/L

TABLE 4. Mean (± s.d.) water quality characteristics in GTA SWP 1997 sediment bioassays.

Test Organism: Mayfly (Hexagenia limbata) [a]			Test Temperature: 19.8°C (0.7)		
Station	*pH*	*D.O.* mg/L	*Conductivity* umho/cm	*Total Ammonia* mg/L	*Un-ionized Ammonia* mg/L
Control	7.96 (.06)	8.8 (0.2)	303 (2)	<0.10 (0)	<0.003
Stn T - 1	8.33 (.14)	8.6 (0.2)	648 (107)	0.31 (0.23)	0.018
Stn T - 2	8.40 (.10)	8.8 (0.2)	1012 (181)	<0.10 (0)	0.010
Stn T - 3	8.37 (.17)	8.8 (0.2)	801 (146)	0.15 (0.10)	0.010
Stn T - 4	8.23 (.03)	8.6 (0.2)	531 (138)	0.26 (0.18)	0.018
Stn T - 5	8.30 (.08)	8.8 (0.1)	521 (78)	1.10 (1.18)	0.097
Stn T - 6	8.32 (.07)	8.8 (0.1)	578 (109)	0.54 (0.53)	0.040
Test Organism: Midge (Chironomus tentans) [b]			Test Temperature: 19.8°C (1.0)		
Station	*pH*	*D.O.* mg/L	*Conductivity* umho/cm	*Total Ammonia* mg/L	*Un-ionized Ammonia* mg/L
Control	7.96 (.06)	8.8 (0.1)	318 (14)	0.12 (0.03)	0.004
Stn T - 1	8.29 (.19)	8.6 (0.1)	621 (106)	0.52 (0.37)	0.045
Stn T - 2	8.39 (.10)	8.8 (0.2)	946 (153)	0.34 (0.22)	0.036
Stn T - 3	8.21 (.10)	8.7 (0.6)	731 (105)	0.29 (0.22)	0.019
Stn T - 4	8.23 (.06)	8.8 (0.1)	444 (43)	0.34 (0.23)	0.022
Stn T - 5	8.24 (.19)	8.7 (0.3)	506 (68)	1.53 (1.35)	0.094
Stn T - 6	8.27 (.07)	8.7 (0.1)	488 (46)	0.73 (0.70)	0.055
Test Organism: Minnow (Pimephales promelas) [a]			Test Temperature: 20.0°C (0.6)		
Station	*pH*	*D.O.* mg/L	*Conductivity* umho/cm	*Total Ammonia* mg/L	*Un-ionized Ammonia* mg/L
Control	7.73 (.10)	8.1 (0.2)	337 (25)	0.32 (0.27)	0.007
Stn T - 1	8.23 (.07)	8.3 (0.2)	705 (126)	1.50 (2.07)	0.099
Stn T - 2	8.16 (.03)	7.9 (0.4)	1134 (233)	0.95 (1.32)	0.062
Stn T - 3	8.01 (.14)	8.0 (0.4)	859 (153)	1.35 (1.84)	0.051
Stn T - 4	7.96 (.25)	7.2 (1.7)	542 (102)	1.47 (2.08)	0.048
Stn T - 5	8.08 (.12)	7.8 (0.8)	551 (80)	2.65 (3.29)	0.163
Stn T - 6	7.98 (.11)	7.9 (0.3)	685 (162)	3.02 (2.70)	0.114

[a] Sample size N=4; [b] Sample size N=3;

Underlining indicates un-ionized ammonia concentrations that exceed the PWQO of 0.02 mg/L

Physical and Nutrient Properties

Sediments were characterized for % sand (2mm-62µm), % silt (62-3.7µm), % clay (3.7-0.1µm), % loss on ignition (LOI), total organic carbon (TOC), total phosphorus (TP) and total Kjeldahl nitrogen (TKN) (Tables 5 and 6).

The Guelph SWP sediments represented a gradation of loam substrates (Millar *et al.*, 1965). About one-half of the test sediments were fine-textured (73 - 75% fines) and classified as clay loam and most of the remaining test sediments were comprised of slightly lower amounts of fine-sized particles (65 - 74%). Only station G-20 was classified as sandy loam due to the high sand content of 64%. Each of the test sediments had moderate amounts of organic matter and varied from 22 mg/g to 50 mg/g TOC. No exceedences of TP or TKN above the SEL were found. Different forms of vegetation were found in all the test samples and was removed during the sieving process just prior to testing. Most of the samples contained varying quantities of rocks and small stones. A few samples (stations G-1, -3, -9 and -20) had an unidentified odour that may have been indicative of hydrogen sulphide gas.

The majority of the GTA SWP sediments were classified as clay loam. Station T-4 had equal portions of sand and silt-sized particles and categorized as loam, while station T-5 was devoid of any sand and was classified as clay. Generally, the sediments contained low amounts of organic material (TOC: 6 - 19 mg/g). Among the test sediments, station T-3 was the most organically-enriched (32 mg/g TOC) and nutrient-enriched (2.8 mg/g TP). The TP sediment concentration was just above the PSGQ-SEL concentration. Detrital material such as aquatic vegetation e.g. cladophora, and stones, were found in most of the test sediments.

Trace Metal Sediment Concentrations

Bulk sediment samples were analysed for 11 trace metals (Tables 7 and 8). The sediment metal concentrations were compared to Severe Effect Level (SEL) and Lowest Effect Level (LEL) concentrations as outlined in the Provincial Sediment Quality Guidelines (PSQGs) (Persaud *et al.*, 1992). The SEL is defined as that chemical concentration in the sediment that could be detrimental to the majority of the macrobenthos and the LEL is the sediment contaminant concentration which can be tolerated by most benthic species.

For the Guelph sediments, 8 of 10 trace metals were commonly measured above PSQG-LEL concentrations. Cadmium, copper, lead and zinc were measured at concentrations over twice their respective LEL at 66% of the sites. Among these four metals, only Cu and Zn were measured at levels above those typically found in Great Lakes nearshore sediments (Ankley *et al.*, 1994; Persaud *et al.*, 1992). The negative control sediment collected from Georgian Bay exemplified such naturally-occurring background metal levels. Copper and Zn sediment concentrations remained well below the SEL concentrations.

A single exceedence above the SEL was reported for the GTA test sediments. Chromium sediment concentration of 790 µg/g at station T-3 was 7-fold higher than the SEL concentration of 110 µg/g. All other trace metals were measured within twice their respective LEL concentration and of a similar order of magnitude compared to those levels found in the negative control sediment.

TABLE 5. Sediment physical and nutrient characteristics in control and Guelph stormwater pond 1997 sediment used in sediment bioassays.

Station	*% Sand* (2mm-62um)	*% Silt* (62-3.7um)	*% Clay* (3.7-0.1um)	*% LOI*	*TOC* mg/g	*TP* mg/g	*TKN* mg/g
Georgian Bay Control	33.0	44.0	22.3	8.6	40	1.0	3.1
Guelph SWP Station G - 1	25.0	45.2	29.4	7.9	44	0.9	2.7
Station G - 3	27.0	40.8	32.0	5.4	42	0.5	1.9
Station G - 4	26.0	47.9	26.4	7.7	46	0.7	2.6
Station G - 5	35.0	45.3	19.5	5.8	32	0.5	1.9
Station G - 6	27.0	40.6	33.1	5.4	43	0.5	1.8
Station G - 7	26.0	41.1	32.6	4.4	22	0.6	1.8
Station G - 8	35.0	46.4	18.8	4.6	26	0.5	1.9
Station G - 9	35.0	41.6	23.2	6.7	50	0.4	2.2
Station G - 20	64.0	23.7	12.1	2.5	24	0.4	0.4 <T
PSQG SEL Conc (mg/g dry weight)					100	2.0	4.8

<T - Trace Amount.

TABLE 6. Sediment physical and nutrient characteristics in control and GTA stormwater pond 1997 sediment used in sediment bioassays.

Station	*% Sand* (2mm-62um)	*% Silt* (62-3.7um)	*% Clay* (3.7-0.1um)	*% LOI*	*TOC* mg/g	*TP* mg/g	*TKN* mg/g
Georgian Bay Control	33.0	44.0	22.3	8.6	40	1.0	3.1
GTA SWP Station T - 1	35.0	30.7	33.8	4.1	19	0.9	1.2
Station T - 2	34.0	37.8	28.3	4.0	13	0.8	1.3
Station T - 3	33.0	37.6	29.9	6.3	32	2.8	1.0
Station T - 4	43.0	41.2	15.6	1.2	6	0.6	0.1 <W
Station T - 5	1.0 <W	39.4	59.6	2.7	11	0.6	0.6
Station T - 6	34.0	33.9	32.0	4.3	17	0.8	1.1
PSQG SEL Conc (mg/g dry weight)					100	2.0	4.8

<W - Not Detected.

Shading indicate sediment nutrient concentrations that exceed PSQG-SELs.

TABLE 7. Bulk concentrations of trace metals in control and Guelph stormwater pond 1997 sediment (µg/g dry weight) used in sediment bioassays.

Station	*Al %*	*As*	*Cd*	*Cr*	*Cu*	*Fe %*	*Hg*	*Mn*	*Ni*	*Pb*	*Zn*
Georgian Bay Control	2.2	5.0	1.4	44	24	3.7	0.09	980	35	54	140
Guelph SWP											
Station G - 1	1.8	9.6	1.9	28	56	2.1	0.09	690	21	93	540
Station G - 3	1.4	5.7	1.4	20	33	1.6	0.07	560	15	54	360
Station G - 4	1.4	7.0	2.3	32	62	1.8	0.08	450	18	96	480
Station G - 5	1.0	5.3	1.5	28	36	1.4	0.08	320	14	72	360
Station G - 6	1.4	6.3	1.7	24	41	1.8	0.06	650	17	75	390
Station G - 7	1.6	6.5	1.4	24	41	1.9	0.07	550	19	75	370
Station G - 8	0.9	3.5	1.1	83	27	1.3	0.05	270	46	49	250
Station G - 9	0.9	6.8	2.2	28	64	1.4	0.06	380	16	120	420
Station G - 20	0.6	2.9	0.8 <T	17	18	0.9	0.05	390	7	34	140
PSQG SEL Conc.	NA	33	10	110	110	4.0	2.0	1100	75	250	820
PSQG LEL Conc.	NA	6.0	0.6	26	16	2.0	0.20	460	16	31	120

<T - Trace Amount; NA- Not Available; Underlining indicate sediment trace metal concentrations that exceed PSQG-LELs.

TABLE 8. Bulk concentrations of trace metals in control and GTA stormwater pond 1997 sediment (µg/g dry weight) used in sediment bioassays.

Station	*Al %*	*As*	*Cd*	*Cr*	*Cu*	*Fe %*	*Hg*	*Mn*	*Ni*	*Pb*	*Zn*
Georgian Bay Control	2.2	5.0	1.4	44	24	3.7	0.09	980	35	54	140
GTA SWP Station T - 1	2.0	3.4	1.0	32	33	2.5	0.05	480	27	32	110
Station T - 2	1.4	2.4	0.5 <T	24	20	1.8	0.04 <T	480	15	20	78
Station T - 3	1.6	3.4	1.4	790	56	2.2	0.08	520	22	43	260
Station T - 4	0.5	1.7	0.3 <T	11	12	1.0	0.03 <T	260	6	6 <T	38
Station T - 5	2.2	4.7	1.0	34	33	3.0	0.04 <T	560	30	25	94
Station T - 6	1.7	3.7	0.8 <T	28	25	2.4	0.04 <T	600	24	23	81
PSQG SEL Conc.	NA	33	10	110	110	4.0	2.0	1100	75	250	820
PSQG LEL Conc.	NA	6.0	0.6	26	16	2.0	0.20	460	16	31	120

<T - Trace Amount; Shading indicate sediment trace metal concentrations that exceed PSQG-SELs.

Underlining indicate sediment trace metal concentrations that exceed PSQG-LELs; NA - Not Available.

Organic Chemical Sediment Concentrations

Concentrations of 19 organochlorine pesticides and 13 chlorinated organic compounds (including total PCBs) in the Guelph SWP test sediments were below the respective detection limits (Tables 9 and 10). Trace amounts of pp-DDE (2 ng/g <T) and hexachlorobenzene (2 ng/g <T) were found at stations G-5 and G-20. Station G-4 had a measurable amount of pp-DDE at a concentration of 14 ng/g. Field sediment was analyzed for oils and greases and bulk sediment concentrations ranged from 1.26 mg/g to 7.75 mg/g and averaged 3.35 mg/g on a sample size of nine (Table 10). Sediment total PAH concentrations ranged from 1,741 ng/g to 51,102 ng/g. Exceedences above the LEL concentration occurred at 77% of the test sites. The highest total PAH concentration at station G-9 was 9-times lower than the calculated SEL concentration (corrected for TOC). Benzo[b]fluoranthene, fluoranthene and pyrene were the most abundant PAH compounds in each of the test sediments and comprised approximately 50% of the total PAH concentration. In addition, no single individual PAH comprised more than 19% of the sum total PAH concentration.

The GTA SWP test sediments had non-detect chemical concentrations for 16 of 20 pesticide compounds and 13 chlorinated organic compounds (excluding total PCBs) (Table 11). Those compounds measured at trace concentrations for some sites included pp-DDE, pp-DDD, pp-DDT, g-chlordane and total PCBs. Each field sediment contained measurable amounts of oils and greases (Range: 0.70 to 13.20 mg/g) (Table 12). The highest oils and greases sediment concentration occurred at station T-3, which coincidentally was the only sample that contained visible amounts of oil and had a petroleum-like odour. Similar PAH concentrations were found among sites (0.4 to 3.0 µg/g) with several PAH compounds found at trace and non-detect concentrations at each site (Table 12). Total PAH sediment concentrations were below the LEL concentration of 4,000 ng/g at all sites.

3.3 Mayfly (*Hexagenia limbata*) 21-day Lethality and Growth Results

The biological data for the two endpoints, mortality and growth, are summarized in Tables 13 and 14, for Guelph and GTA sites, accordingly. In the Guelph study, percent mortality was nil in the control and all test sediments. Mayfly body weights varied significantly among test and control sediments (ANOVA; $p<0.0001$) (Figure 1). Among the test sediments, stations G-1, -4, -5, -6, -9 exhibited the best growth. Final weights were at least 5-fold greater in size in relation to the initial starting weight of 4.2 mg. Animals retrieved from three other test sites (stations G-3, -7, -20) were significantly smaller than the above group of test sites but still illustrated an acceptable doubling in weight as compared to either the negative control or the starting weight. Only station G-8 sediment resulted in a lower degree of mayfly growth of 10.3 mg.

In the GTA study, mayfly mortality was significantly different among sediments (ANOVA; $p<0.0001$). Percent mortality for station T-3 of 26% was statistically higher than the other test sediments (Range: 0% to 3% mortality; Tukey's multiple range test) and the negative control exposure (0% mortality; Dunnett's *t*-test). Mayfly survival among the other test sediments were similar to the control. The growth data differentiated the test sediments into three significant groups (ANOVA;$p<0.0001$) (Figure 2). Above average growth was reported for stations T-4 and T-5, intermediate growth for stations T-1, -2 and -6, and substantially lower growth occurred for station

TABLE 9. Bulk sediment concentrations for chlorinated organics and pesticides in Guelph stormwater pond 1997 sediment (ng/g, dry weight) used in bioassays.

All Stations (exceptions listed below)	Heptachlor	1 <W
	Aldrin	1 <W
	Mirex	5 <W
	a-BHC	1 <W
	b-BHC	1 <W
	g-BHC	1 <W
	a-Chlordane	2 <W
	g-Chlordane	2 <W
	Oxychlordane	2 <W
	op-DDT	5 <W
	pp-DDD	5 <W
	pp-DDT	5 <W
	pp-DDE	1 <W
	Methoxychlor	5 <W
	Heptachlor epoxide	1 <W
	Endosulphan I	2 <W
	Dieldrin	2 <W
	Endrin	4 <W
	Endosulphan II	4 <W
	Endosulphan sulphate	4 <W
	Hexachlorobutadiene	1 <W
	Octachlorostyrene	1 <W
	Hexachlorobenzene	1 <W
	123-Trichlorobenzene	2 <W
	124-Trichlorobenzene	2 <W
	135-Trichlorobenzene	2 <W
	1234-Tetrachlorobenzene	1 <W
	1235-Tetrachlorobenzene	1 <W
	1245-Tetrachlorobenzene	1 <W
	Hexachloroethane	1 <W
	Pentachlorobenzene	1 <W
	236-Trichlorotoluene	1 <W
	245-Trichlorotoluene	1 <W
Station G - 4	pp-DDE	14.0
Station G - 5	pp-DDE	2.0 <T
Station G - 20	Hexachlorobenzene	2.0 <T

<W - Not Detected; <T - Trace Amount.

TABLE 10. Bulk concentrations of polycyclic aromatic hydrocarbons, total PCBs and oils and greases in control and Guelph stormwater pond 1997 sediment used in sediment bioassays or on field samples.

Parameter	Control	Stn G -1	Stn G - 3	Stn G - 4	Stn G - 5	Stn G - 6	Stn G - 7	Stn G - 8	Stn G - 9	Stn G - 20
Acenaphthene	40 <W	20 <W	20 <W	37 <T	73	20 <W	20 <W	34 <T	160	20 <W
Acenaphythylene	40 <W	20 <W	20 <W	20 <W	20 <W	20 <W	20 <W	20 <W	34 <T	20 <W
Anthracene	40 <W	30 <T	33 <T	85	140	20 <W	20 <W	73	350	25 <T
Benzo[a]anthracene	40 <W	250	340	860	1100	100	240	610	2800	160
Benzo[b]fluoranthene	88 <T	780	1100	2100	2700	250	680	2100	7100	400
Benzo[k]fluoranthene	44 <T	230	370	660	850	78	160	560	1800	100
Benzo[ghi]perylene	83 <T	410	620	1000	1200	140	320	960	2800	180
Benzo[a]pyrene	40 <W	410	610	1200	1600	120	340	1200	4300	240
Chrysene	40 <W	570	790	1400	1800	160	360	1300	4600	290
Dibenzo[ah]anthracene	80 <W	70 <T	100	180	230	40 <W	55 <T	190	660	40 <W
Fluoranthene	60 <T	1100	1500	2700	3900	250	760	2500	9400	510
Fluorene	40 <W	37 <T	31 <T	94	140	20 <W	20 <W	56	270	20 <W
Indeno[123-cd]pyrene	110 <T	460	650	1200	1500	150	370	1100	3600	200
Naphthalene	40 <W	20 <W	20 <W	20 <W	20 <W	20 <W	20 <W	20 <W	28 <T	20 <W
Phenanthrene	40 <W	410	460	1000	1800	83	240	870	3600	230
Pyrene	54 <T	1000	1400	2900	3900	270	710	2500	9600	550
Total PAHs (ng/g, dry weight)	879 <T	5817	8064	15456	20973	1741	4335	14093	51102	3005
Total PCBs (ng/g, dry weight)	20 <W	20 <W	20 <W	20 <W	20 <W	20 <W	20 <W	20 <W	20 <W	20 <W
Oils & Greases (mg/g, dry weight)	NA	1.53	7.75	1.67	1.96	2.04	4.40	5.56	1.26	4.03

<W - Not Detected; T - Trace Amount Measured; NA - Not Available; PSQG's not available for aceapthene, acenapthylene, benzo[b]flourene and naphthalene.
Underlining indicate sediment PAH concentrations that exceed PSQG-LELs and oils and greases sediment concentrations above the open water disposal guideline.

TABLE 11. Bulk sediment concentrations for chlorinated organics and pesticides in GTA stormwater pond 1997 sediment (ng/g, dry weight) used in bioassays.

All Stations (exceptions listed below)	Heptachlor	1 <W
	Aldrin	1 <W
	Mirex	5 <W
	a-BHC	1 <W
	b-BHC	1 <W
	g-BHC	1 <W
	a-Chlordane	2 <W
	g-Chlordane	2 <W
	Oxychlordane	2 <W
	op-DDT	5 <W
	pp-DDD	5 <W
	pp-DDT	5 <W
	pp-DDE	1 <W
	Methoxychlor	5 <W
	Heptachlor epoxide	1 <W
	Endosulphan I	2 <W
	Dieldrin	2 <W
	Endrin	4 <W
	Endosulphan II	4 <W
	Endosulphan sulphate	4 <W
	Hexachlorobutadiene	1 <W
	Octachlorostyrene	1 <W
	Hexachlorobenzene	1 <W
	123-Trichlorobenzene	2 <W
	124-Trichlorobenzene	2 <W
	135-Trichlorobenzene	2 <W
	1234-Tetrachlorobenzene	1 <W
	1235-Tetrachlorobenzene	1 <W
	1245-Tetrachlorobenzene	1 <W
	Hexachloroethane	1 <W
	Pentachlorobenzene	1 <W
	236-Trichlorotoluene	1 <W
	245-Trichlorotoluene	1 <W
Station T - 2	pp-DDE	2.0 <T
Station T - 3	pp-DDE	2.0 <T
	pp-DDD	10.0 <T
	pp-DDT	15.0 <T
Station T - 4	pp-DDE	2.0 <T
	g-Chlordane	4.0 <T

<W - Not Detected; <T - Trace Amount.

TABLE 12. Bulk concentrations of polycyclic aromatic hydrocarbons, total PCBs and oils and greases in control and GTA stormwater pond 1997 sediment used in sediment bioassays or on field samples.

Parameter	Control	Stn T - 1	Stn T - 2	Stn T - 3	Stn T - 4	Stn T - 5	Stn T - 6
Acenaphthene	40 <W	20 <W	20 <W	50	20 <W	20 <W	20 <W
Acenaphythylene	40 <W	20 <W	20 <W	20 <W	20 <W	20 <W	20 <W
Anthracene	40 <W	20 <W	21 <T	91	20 <W	20 <W	20 <W
Benzo[a]anthracene	40 <W	160	210	150	86	120	60
Benzo[b]fluoranthene	88 <T	210	370	260	120	100	34 <T
Benzo[k]fluoranthene	44 <T	79	140	80	42	38 <T	20 <W
Benzo[ghi]perylene	83 <T	140	210	170	75 <T	72 <T	40 <W
Benzo[a]pyrene	40 <W	120	230	110	77	59	20 <W
Chrysene	40 <W	120	260	280	58	74	20 <W
Dibenzo[ah]anthracene	80 <W	40 <W	40 <W	40 <W	40 <W	40 <W	40 <W
Fluoranthene	60 <T	320	550	270	160	160	32 <T
Fluorene	40 <W	20 <W	20 <W	110	20 <W	20 <W	20 <W
Indeno[123-cd]pyrene	110 <T	130	220	110	80	75 <T	40 <W
Naphthalene	40 <W	20 <W	20 <W	88	20 <W	20 <W	20 <W
Phenanthrene	40 <W	110	230	470	37 <T	82	20 <W
Pyrene	54 <T	280	480	480	140	160	39 <T
Total PAHs (ng/g, dry weight)	879 <T	1809	3041	2779	1015	1080	465 <T
Total PCBs (ng/g, dry weight)	20 <W	40	20 <W	40	20 <W	40	20 <W
Oils & Greases (mg/g, dry weight)	NA	0.71	1.85	13.20	0.69	0.70	3.46

<W - Not Detected; T - Trace Amount Measured; NA - Not Available.

Underlining indicate sediment PAH concentrations that exceed PSQG-LELs and oils and greases sediment concentrations above the open water disposal guideline.

PSQG's not available for acenapthene, acenapthylene, benzo[b]fluourene and naphthalene.

TABLE 13. Summary of biological results on mayfly, midge and minnow sediment bioassays for control and Guelph stormwater pond 1997 sediments.

Mean values (± standard deviation) where sample size n=3 replicates.

Test Organism	*Hexagenia limbata* (Mayfly)		*Chironomus tentans* (Midge)		*Pimephales promelas* (Fathead Minnow)
Station	*% Mortality*	*Ave. Individual Body Weight* (mg wet wt.)	*% Mortality*	*Ave. Individual Body Weight* (mg wet wt.)	*% Mortality*
Honey Harbour Control	A 0 (0)	E 6.72 (0.5)	A 6.6 (7)	B 10.77 (1.3)	A 0 (0)
Guelph Station G - 1	A 0 (0)	B 23.35 (4.0)	A 0 (0)	AB 11.98 (1.3)	A 0 (0)
Station G - 3	A 0 (0)	CD 13.86 (2.8)	A 0 (0)	A 13.81 (2.0)	A 0 (0)
Station G - 4	A 0 (0)	A 38.20 (0.8)	A 2.2 (4)	A 12.96 (0.8)	A 36.6 (55)
Station G - 5	A 0 (0)	B 27.72 (3.6)	A 0 (0)	A 13.32 (0.5)	A 0 (0)
Station G - 6	A 0 (0)	B 28.65 (1.9)	A 0 (0)	AB 12.00 (0.6)	A 0 (0)
Station G - 7	A 0 (0)	CD 13.34 (1.9)	A 4.4 (8)	AB 12.48 (0.3)	A 0 (0)
Station G - 8	A 0 (0)	DE 10.36 (1.1)	A 2.2 (4)	AB 12.38 (1.5)	A 0 (0)
Station G - 9	A 0 (0)	B 25.00 (1.0)	A 0 (0)	A 13.69 (0.3)	A 6.6 (11)
Station G - 20	A 0 (0)	C 16.53 (2.5)	A 2.2 (4)	A 12.94 (0.9)	A 0 (0)
% MSD	-	-	8.1	-	37.7
% C.V.	-	9.5	183.3	7.0	152.0
D.P.	-	13.2	2.0	2.0	5.0

* %Mortality value is significantly different than the control sediment (Dunnett's 1-tailed t-test;$p<0.05$).

A Means sharing a common letter within a column are not significantly different; Tukey's HSD test for % Mortality ($p<0.05$) and planned comparisons using LSMEANS for comparing Body Weight ($p<0.01$).

MSD - Minimum Significant Difference; C.V. - Coefficient of Variation; D.P. - Discriminatory Power.

TABLE 14. Summary of biological results on mayfly, midge and minnow sediment bioassays for control and GTA stormwater pond 1997 sediments.

Mean values (± standard deviation) where sample size n=3 replicates.

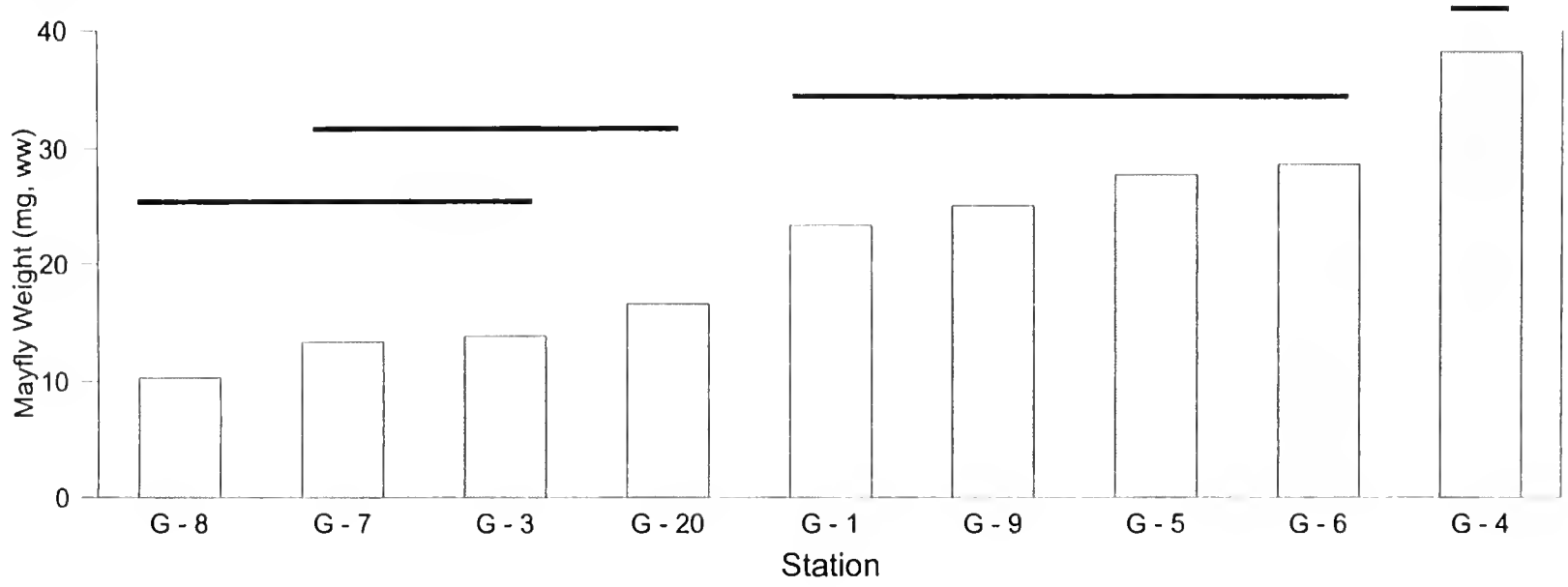
Test Organism	*Hexagenia limbata* (Mayfly)		*Chironomus tentans* (Midge)		*Pimephales promelas* (Fathead Minnow)
Station	*% Mortality*	*Ave. Individual Body Weight* (mg wet wt.)	*% Mortality*	*Ave. Individual Body Weight* (mg wet wt.)	*% Mortality*
Honey Harbour Control	A 0 (0)	C 5.98 (0.9)	AB 2.2 (4)	BC 8.75 (1.4)	A 4.1 (7)
GTA Station T - 1	A 0 (0)	B 16.85 (2.1)	AB 11.1 (19)	A 12.15 (2.2)	A 4.1 (7)
Station T - 2	A 3.3 (6)	B 16.21 (1.8)	A 0 (0)	C 6.77 (1.9)	A 16.6 (29)
Station T - 3	B * 26.6 (15)	C 6.70 (1.4)	AB 15.5 (4)	D 2.73 (0.4)	A 12.5 (22)
Station T - 4	A 0 (0)	A 21.91 (1.5)	B * 33.3 (29)	AB 9.34 (1.8)	A 0 (0)
Station T - 5	A 0 (0)	A 24.67 (4.2)	AB 2.2 (4)	A 11.21 (0.5)	A 16.6 (19)
Station T - 6	A 0 (0)	B 14.75 (1.2)	AB 4.4 (4)	AB 10.70 (2.4)	A 0 (0)
% MSD % C.V. D.P.	12.7 71.4 7.6	- 11.8 8.9	27.8 90.9 3.3	- 17.3 6.1	32.8 154.2 1.2

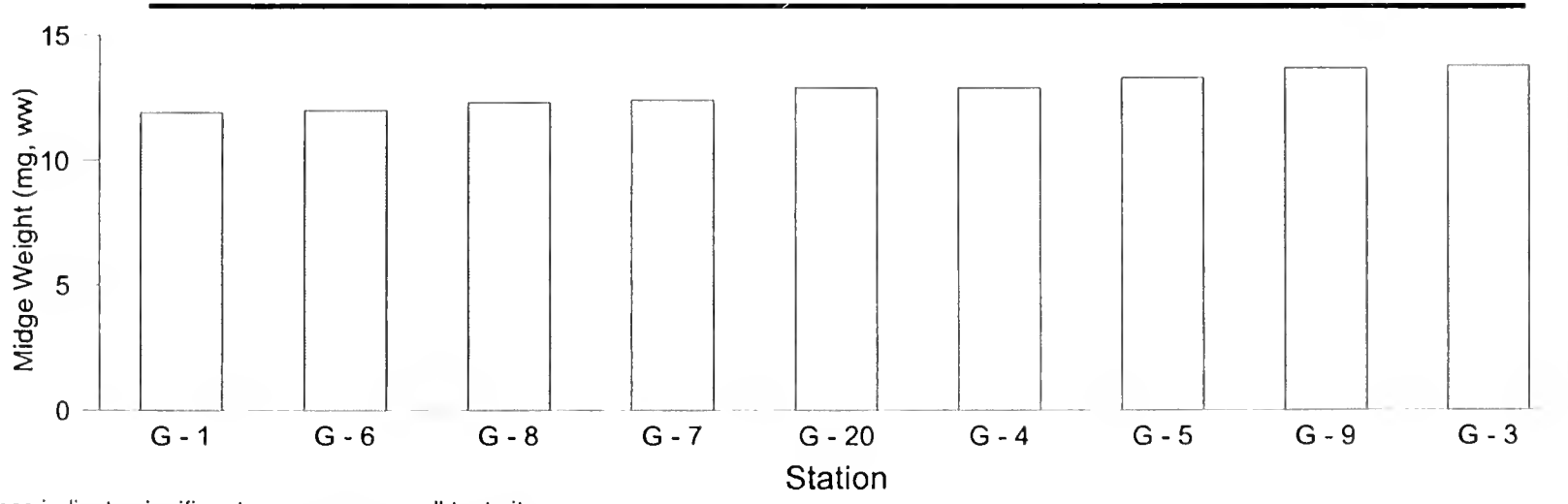
* %Mortality value is significantly different than the control sediment (Dunnett's 1-tailed t-test;$p<0.05$).

A Means sharing a common letter within a column are not significantly different; Tukey's HSD test for % Mortality ($p<0.05$) and planned comparisons using LSMEANS for comparing Body Weight ($p<0.01$).

MSD - Minimum Significant Difference; C.V. - Coefficient of Variation; D.P. - Discriminatory Power.

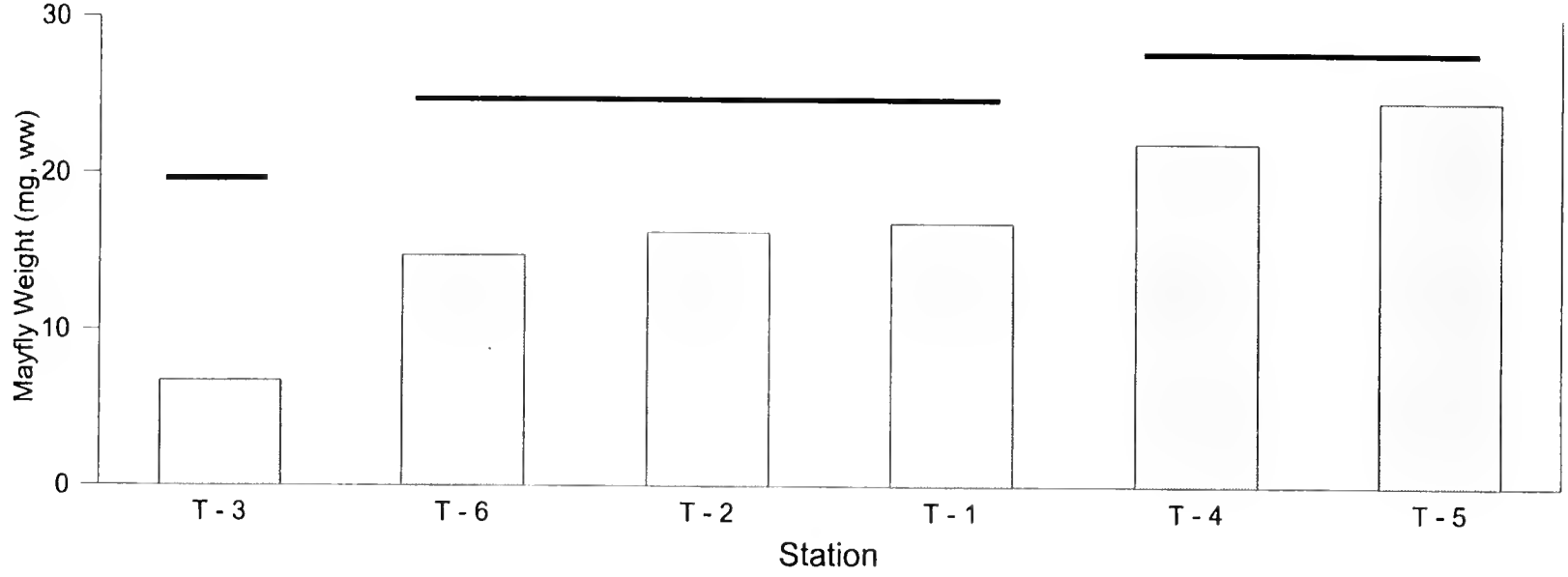
Figure 1. Benthic Growth Effects for Guelph SWP Sediments

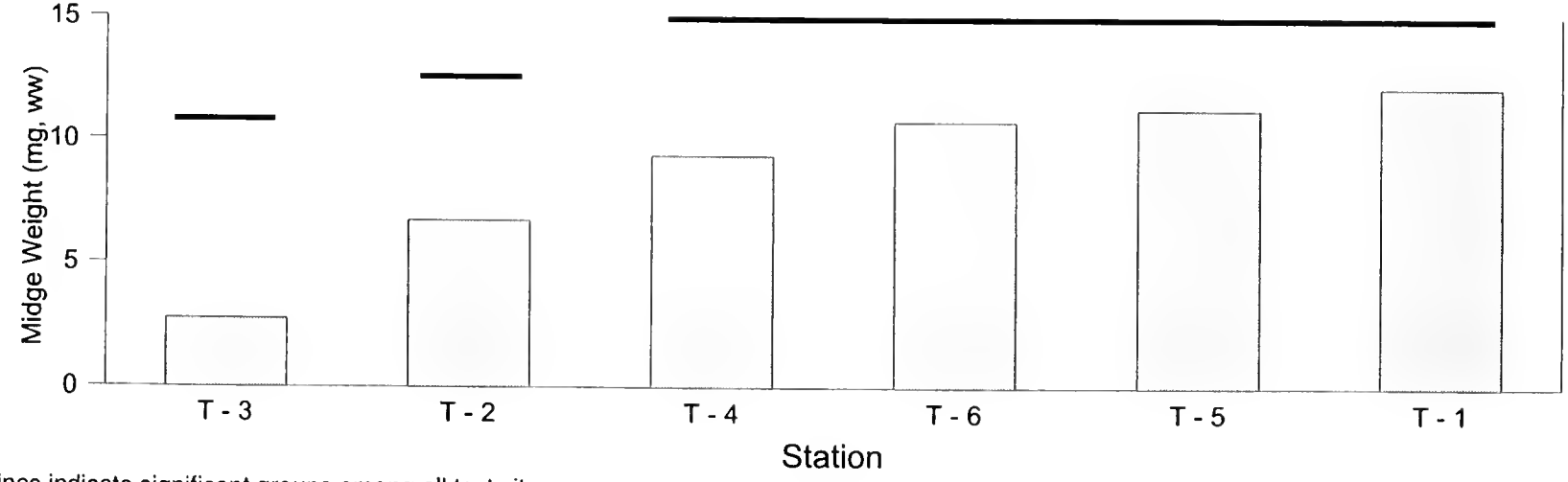




Lines indicate significant groups among all test sites.

Figure 2. Benthic Growth Effects for GTA SWP Sediments





Lines indicate significant groups among all test sites.

T-3. The average body weight of 6.7 mg reported for station T-3 was at least 54% less than the mayfly weight attained at all other test sites and is indicative of impaired growth, even though it was comparable to the negative control weight.

The relatively low body weight of <7mg found for the negative control animals is likely attributed to the longer sediment storage time (~12 months) versus the freshly collected (~ 3 weeks) test sediments in both studies. The prolonged storage may have altered the nutritive value of the control sediment, thereby affecting organism feeding rates (Boese *et al.*, 1996). Mayfly growth depends on the quality and quantity of detrital material found in the sediment, since a supplemental diet was not provided during the bioassay. The inclusion of an appropriate reference control sediment would have improved the interpretation of the mayfly growth data by having a more similar substrate type as the test sediments, subjected to similar storage conditions, and thereby being a better indicator of mayfly growth capability. For these reasons, a greater emphasis was placed on describing any among test site differences.

3.4 Chironomid (*Chironomus tentans*) 10-day Lethality and Growth Results

Results for chironomid growth and lethality are reported in Tables 13 and 14 and growth data is depicted in Figures 1 and 2. No significant differences in midge mortality was found among the Guelph SWP sediments (ANOVA; $p<0.44$). Chironomid mortality averaged 0% to 4% for the test sediments and 6% in the negative control sediment. Each test sediment received a similar ranking using Tukey's multiple range test. Midge growth among the test sediments (11.9 mg to 13.8 mg) did not vary appreciably and were either equal to or above the weight attained in the negative control exposure. The negative control weight of 10.7 mg is considered to be a good rate of midge growth.

There were no measurable differences in midge mortality among GTA SWP sediments relative to the negative control using Tukey's multiple range test. Most test sediments had percent mortality less than 15% which is well below the maximum acceptable criterion of 25% mortality. Station T-4 was the exception, the percent mortality of 33% was found to be higher than the control mortality using the paired Dunnett's *t*-test. However, the associated degree of error was high due to variability in mortality observed among replicate samples. This level of uncertainty resulted in a similar ranking to the negative control when the less conservative comparative statistical test was employed. Midge growth was significantly reduced for station T-3 relative to all other test sites and the negative control (ANOVA; $p<0.0001$). The body weight (2.7 mg) was 68% lower than the control weight of 8.7 mg. The other test sediments promoted midge growth comparable to or better than that found in the negative control sediment.

3.5 Fathead Minnow (*Pimephales promelas*) 21-day Lethality Results

Juvenile fathead minnow percent mortality data is reported in Tables 13 and 14, for the Guelph and GTA studies, respectively. Control survival was 100%, along with seven test exposures in the Guelph study. Station G-9 percent mortality was 6% and station G-4 incurred the highest percent mortality of 36%. Neither value was significantly different than the control and other test sediments (ANOVA; $p<0.18$). Unequal losses were recorded among the replicate jars

for station G-4 e.g. Replicate A - 100%, Replicate B - 10%, Replicate C - 0%. The reason for the complete loss of animals in one of the three G-4 test jars from Day-11 to Day-17 was not readily apparent. This isolated incident suggests the possibility of an injured or diseased individual that rapidly infected all other fish.

Minnow percent mortality for the GTA test sediments ranged from 0% to 16% with no significant differences found either among sites nor relative to the control (4% mortality) (ANOVA; $p<0.69$). According to daily laboratory observations, all mortalities occurred within the first 48 hours and, therefore, were attributed to handling stress.

3.6 Quality Assurance Data

An evaluation was made on both data sets in order to determine the repeatability of the test results (Tables 13 and 14). Test precision was consistently better using the sublethal benthic endpoints as compared to the corresponding lethality results and applied to both studies. Mayfly growth and midge growth showed excellent test precision with % coefficient of variation falling under 20%. Relative differences in species sensitivity was noted, particularly for the Guelph study. The ability to detect differences in biological response was improved using *Hexagenia* which had the highest discriminatory power (D.P.=13 and 8). Mayfly survival, midge survival, midge growth and minnow survival were less sensitive and failed to detect any important differences among sites.

The MSD or minimum significant difference is a quantitative measure of test precision which describes the ability to detect a significant effect in the paired response between the control versus the test sample. Among the lethality results, the mayfly assay appeared to be more effective in detecting small differences in lethality among sites, as reflected in the excellent MSD value of 12% reported for the GTA study. This is in agreement with the average MSD that was calculated from ten previous OMOE mayfly toxicity tests e.g. MSD=12% (D. Bedard, OMOE, unpublished data). No determination was possible for the Guelph study due to the absence of an effect. The minnow tests and the GTA midge test failed to meet the same quality test standards based on past laboratory performance. The statistical power was reduced due to the level of variability in organism mortality recorded at certain test sites. The need to use a smaller sample size, e.g. 8 fish per replicate, in the GTA minnow assay also contributed to the higher MSD. Nevertheless, each of the GTA test sites were still characterized as being non-toxic to fathead minnows with average mortality equal to or less than the accepted control mortality criterion of 15% mortality.

The 48-hour copper LC50 (95% C.I.) for the water-only reference toxicant exposures for *H. limbata* for the Guelph and GTA studies were 1.83 (0.91 - 7.96) mg/L and 1.16 (0.82 - 1.83) mg/L, respectively. These values were within their respective acceptable 48-h LC50 (± 2 s.d.) range of 1.15 (1.34) mg/L and 1.14 (1.36) mg/L, according to a previous series of reference toxicant tests. Similarly, for *C. tentans*, the LC50s were 0.83 (0.85 - 1.23) mg/L and 0.76 (0.62 - 0.94) mg/L, as compared to an expected 48-h LC50 (± 2 s.d.) of 1.26 (0.94) mg/L and 1.21 (0.91) mg/L, for the Guelph and GTA tests, accordingly. This indicates that the relative sensitivity of each batch of the test organisms fell within a normal response range.

3.7 Chemical Bioaccumulation in *Pimephales promelas*

The examination of chemical availability to aquatic organisms is valuable for assessing the potential for chemical transfer through the food web. The primary objective of this test design is to make general observations on whole organism tissue concentrations as they relate to overall bulk chemical concentrations in the sediment and differences in chemical uptake among sites. Surviving fathead minnows were submitted for the analysis of total PCBs and pesticides. All values are based on whole-body tissue concentrations and reported as wet weight. The average dry to wet weight ratio was 0.16.

The sources of organic compound accumulation to forage fish include direct contact with the sediment and uptake from the overlying water. Factors that control chemical accumulation by forage fish include those that affect chemical adsorption and desorption such as sediment organic content, particle size distribution and the chemical's solubility properties commonly expressed by the octanol-water partition coefficient, Kow (Lake *et al.*, 1990). Biotic factors affecting uptake include metabolism and lipid content (Boese *et al.*, 1995).

Tables 15 and 16 report the total PCB and pesticide tissue concentrations (wet weight) for the Guelph and GTA sediment exposures, respectively. No estimate of within-site variability in tissue concentrations is available due to an analysis of a single pooled sample per station. For this reason, a preliminary qualitative assessment of chemical uptake and availability will be made through the examination of trends in contaminant concentrations. Further comparisons are made between sediment and tissue concentrations for each chemical.

For the Guelph study, most of the organic tissue concentrations represented low-level concentrations that were similar to those measured in the control animals and indicative of tissue concentrations derived through laboratory conditions. Similar tissue concentrations were also found in a group of pre-exposed minnows. Trace quantities of total PCBs (23 ng/g), pp-DDE (4.2 ng/g), pp-DDD (0.8 ng/g) and pp-DDT (2.5 ng/g) were measured in the Honey Harbour control fish. In comparison, total PCBs in most test fish ranged from 13 to 33 ng/g, with the exception of station G-5 (83 ng/g). The source of the PCBs is unclear since all the Guelph SWP sediments had no detectable amounts of PCBs. Pesticide tissue concentrations ranged from 2.1 to 8.5 ng/g for pp-DDE at most sites. A slightly higher pp-DDE concentration was measured at station G-20 of 12.9 ng/g. A quantifiable amount of pp-DDE was measured in G-4 sediment of 14.0 ng/g and resulted in the lowest recorded tissue concentration suggesting the compound is not rapidly bioaccumulative. Trace amounts of pp-DDD (0 - 3.9 ng/g) and pp-DDT (1.0 - 3.4 ng/g) were reported among the Guelph fish samples with little difference noted among sites.

For the GTA study, the control animals had consistently higher concentrations of all four chemicals (total PCBs 69 ng/g; pp-DDE 12.6 ng/g; pp-DDD 3.4 ng/g; pp-DDT 4.5 ng/g), as compared to the control animals in the Guelph study. A different batch of fish were used in each study and differences in tissue concentrations may have arisen from culturing practices. A separate sample of pre-exposed animals were not taken prior to the GTA SWP bioassay. For the most part, fish exposed to the test sediments acquired tissue organic chemical concentrations similar to or less than the control fish tissue concentrations. There were no obvious differences in tissue concentrations among the SWPs. The corresponding sediment pesticide concentrations were either non-detect or at trace concentrations. Total PCB sediment (20 - 40 ng/g dw) and tissue

TABLE 15. Total PCBs, pp-DDE, pp-DDD and pp-DDT concentrations (ng/g, wet wt) in fathead minnows exposed to control and Guelph SWP 1997 sediments in the laboratory.

Station	*Total PCBs* ng/g	*pp-DDE* ng/g	*pp-DDD* ng/g	*pp-DDT* ng/g
Honey Harbour Control	23 <T	4.2 <T	0.8 <T	2.5 <T
Guelph SWP				
Station G - 1	19 <T	2.5 <T	0.0	2.1 <T
Station G - 3	20 <T	3.0 <T	0.0	1.8 <T
Station G - 4	13 <T	2.1 <T	0.0	3.4 <T
Station G - 5	83 <T	4.0 <T	0.0	2.9 <T
Station G - 6	24 <T	3.0 <T	0.0	1.4 <T
Station G - 7	33 <T	2.9 <T	1.3 <T	2.4 <T
Station G - 8	17 <T	8.5 <T	2.8 <T	1.0 <T
Station G - 9	18 <T	2.4 <T	0.7 <T	1.5 <T
Station G - 20	19 <T	12.9 <T	3.9 <T	1.3 <T

Sample size n=1; <T - Trace Amount.

TABLE 16. Total PCBs, pp-DDE, pp-DDD and pp-DDT concentrations (ng/g, wet wt) in fathead minnows exposed to control and GTA SWP 1997 sediments in the laboratory.

Station	*Total PCBs* ng/g	*pp-DDE* ng/g	*pp-DDD* ng/g	*pp-DDT* ng/g
Honey Harbour Control	69	12.6 <T	3.4 <T	4.5 <T
GTA SWP				
Station T - 1	52	9.7 <T	2.6 <T	1.8 <T
Station T - 2	60	6.3 <T	1.5 <T	1.5 <T
Station T - 3	33 <T	8.9 <T	4.7 <T	2.8 <T
Station T - 4	59	11.4 <T	3.6 <T	1.0 <T
Station T - 5	70	19.1 <T	4.1 <T	3.4 <T
Station T - 6	40 <T	5.7 <T	0.8 <T	1.6 <T

Sample size n=1; <T - Trace Amount.

concentrations (33 - 70 ng/g ww) both varied two-fold among sites.

4.0 DISCUSSION

Spatial Trends in Sediment Toxicity and Chemical Bioaccumulation

Site to site differences in sediment quality were identified in order to measure any spatial trends in biological effect for the two independent data sets. This was determined by the magnitude of an effect using statistical test methods and chemical minnow concentrations as they relate to existing federal and provincial tissue guidelines. Each endpoint was considered as being either a significant, toxic (T) or non-significant, non-toxic (N) response (Tables 17 and 18). In addition, the lethality endpoint received a greater weighting over the respective sublethal endpoint, where applicable. The final rating is based on the total number of positive hits recorded for each of the five biological endpoints along with the chemical bioaccumulation data.

Most of the Guelph test sediments were considered to be of a high sediment quality or non-impacted due to the lack of any significant negative biological effect. Station G-8 was the only location that was slightly impaired because of one positive response. This occurred in the *Hexagenia limbata* toxicity test and measured as a poor rate of growth. A similar trend in sediment quality was apparent among the GTA SWP sediments. Five out of six test sediments were non-impaired. Station T-3 was considered impaired due to significant lethal and sublethal effects measured in both benthic toxicity tests.

Statistical differences in fathead minnow tissue concentrations was not carried out due to the lack of sample replication. Instead, trends in pesticide and PCB tissue concentrations were examined. There was no consistent uptake of PCBs or DDT compounds among the SWP sediments, relative to the negative control sediment, in each study. Generally fish body burdens were at or near trace concentrations and corresponded to the low concentrations measured in the sediments. None of the 21-day tissue concentrations reached either the IJC Aquatic Life Guideline (IJC, 1988) nor the provincial sport fish consumption limit (MOE/MNR, 1999). Chemical uptake and bioaccumulation of organic compounds by fathead minnows in short-term exposures did not indicate any differences in sediment quality, for either study.

Relationships Between Biological Endpoints and Sediment Physical/Chemical Characteristics

Spearman rank correlation coefficients were calculated among each test species and endpoint within their respective study (Tables 19 and 20). Overall the test endpoints were not significantly intercorrelated as a result of uniformity in biological effects among sites. In other words, most of the sites were deemed to be non-toxic, regardless of the endpoint being examined. There was a high frequency of significant relationships ($p<1.0$) that occurred between mayfly survival and all other endpoints in the Guelph study. These correlations can be disregarded as being critical, since mayfly survival was 100% across all sites, which subsequently led to the erroneous correlations. The most important correlation was measured between mayfly survival and midge growth in the GTA study ($r=+0.84$; $p<0.05$). This relationship provides multiple lines of evidence that support the impaired status assigned to station T-3.

TABLE 17. Spatial variability in sediment toxicity and sediment quality for Guelph stormwater pond 1997 samples

Station	Sediment Oils & Greases (mg/g dry wt)	Mayfly Mortality	Mayfly Ave wt	Midge Mortality	Midge Ave wt	Minnow Mortality
Guelph SWP						
Stn G - 1	1.5	N	N	N	N	N
Stn G - 4	1.6	N	N	N	N	N
Stn G - 5	1.9	N	N	N	N	N
Stn G - 6	2.0	N	N	N	N	N
Stn G - 9	1.2	N	N	N	N	N
Stn G - 20	4.0	N	N	N	N	N
Stn G - 7	4.4	N	N	N	N	N
Stn G - 3	7.7	N	N	N	N	N
Stn G - 8	5.5	N	T	N	N	N

N - Not Toxic, % mortality less than control criteria or $p>0.05$ and $p>0.10$ for growth data;
T - Toxic, % mortality greater than control criteria or $p<0.05$ and $p<0.10$ for growth data.

TABLE 18. Spatial variability in sediment toxicity and sediment quality for GTA stormwater pond 1997 samples.

Station	Sediment Oils & Greases (mg/g dry wt)	Mayfly Mortality	Mayfly Ave wt	Midge Mortality	Midge Ave wt	Minnow Mortality
GTA SWP						
Stn T - 1	0.7	N	N	N	N	N
Stn T - 2	1.8	N	N	N	N	N
Stn T - 4	0.7	N	N	N	N	N
Stn T - 5	0.7	N	N	N	N	N
Stn T - 6	3.4	N	N	N	N	N
Stn T - 3	13.2	T	T	N	T	N

N - Not Toxic, % mortality less than control criteria or $p>0.05$ and $p>0.10$ for growth data;
T - Toxic, % mortality greater than control criteria or $p<0.05$ and $p<0.10$ for growth data.

TABLE 19. Spearman rank correlation coefficients indicating significant positive (direct) correlations among toxicity data for Guelph SWP 1997 sediments.

	Mayfly Survival	Mayfly Growth	Midge Survival	Midge Growth
Mayfly Growth	+ 1.00 **			
Midge Survival	+ 1.00 **	n.s.		
Midge Growth	+ 1.00 **	n.s.	n.s.	
Minnow Survival	+ 1.00 **	n.s.	n.s.	n.s.

TABLE 20. Spearman rank correlation coefficients indicating significant positive (direct) correlations among toxicity data for GTA SWP 1997 sediments.

	Mayfly Survival	Mayfly Growth	Midge Survival	Midge Growth
Mayfly Growth	n.s.			
Midge Survival	n.s.	n.s.		
Midge Growth	+ .845 **	n.s.	n.s.	
Minnow Survival	n.s.	n.s.	n.s.	n.s.

** $p < 0.05$; * $p < 0.10$; n.s. - Not Significant at $p>0.10$.

The toxicity endpoints were also compared to sediment physical, nutrient and chemical parameters to aid in determining potential causes for the observed laboratory effects (Tables 21 and 22). Correlation analysis was carried out on bulk sediment concentrations for each parameter. Additional correlations were run for those chemicals that significantly co-varied with those sediment characteristics, including TOC and %fines, that may affect chemical availability and distribution. Overall, very few negative correlations were detected in the Guelph study. This is not unexpected given the absence of any adverse biological effects. A number of sediment parameters were associated with minnow survival but not to an extent that elicited significant differences in toxicity. Results from the mayfly bioassay indicated variable nymph growth among sites. This endpoint also had a number of significant correlations both to inorganic (cadmium, zinc) and organic (oils and greases) sediment contamination. Differences in mayfly growth was metal-related but in an inverse manner. In other words, mayfly growth actually increased as the metal sediment concentration increased. Correlation analysis indicated that Cd sediment concentration co-varied with sediment TOC and after correcting for this variable it was noted that mayfly growth no longer correlated with Cd (OC).

The most reliable predictor of mayfly growth was the amount of oils and greases measured in the field sediment. There was a moderate negative relationship, r= -0.68 at p=0.05. The poorest mayfly growth occurred at station G-8 which had a corresponding oils and greases sediment concentration of 5.5 mg/g. This was followed by stations G-3 and G-7 which shared identical rankings and also had some of the highest reported oils and greases sediment concentrations of 7.7 and 4.4 mg/g, respectively. Most of the remaining Guelph test sediments had concentrations under 2.0 mg/g. This pattern in mayfly growth was even better described by oils and greases sediment concentrations normalized for sediment TOC (r= -0.78 at p=0.02). The groupings (according to the Tukey's multiple range test) were as follows: Group A (stations G-1, -4, -5, -6, -9) had sediment oils and greases concentration ranging from 24 mg/g OC to 59 mg/g OC; Group B (stations G-3, -7, -20) with 166 mg/g OC to 200 mg/g OC; and Group C (station G-8) had 211 mg/g OC. The distribution of related organic compounds in the sediment, including PAHs and PCBs, failed to correlate with mayfly growth.

The sediment bioassays conducted on the GTA sites clearly identified station T-3 as having a detrimental impact on mayfly and midge growth and to a lesser extent on mayfly survival. Spearman rank correlation data found significant negative responses for the mayfly endpoints with respect to sediment TOC, oils and greases, chromium and total PAHs. Station T-3 had the highest oils and greases field sediment concentration of 13.2 mg/g. This concentration was 3-fold higher than any of the other test sediments. In addition, Cr sediment concentration surpassed the PSQG-SEL by 7-fold. Station T-3 also had the highest quantity of TOC of 32 mg/g among all GTA SWP sediments.

Other toxicological studies suggest a relatively high Cr sediment concentration is required before an effect is observed and chemical speciation and availability are factors affecting Cr toxicity. Freshwater sediments collected from Tannery Bay in the St. Marys River that are Cr-enriched have been tested using aquatic invertebrates. In a study using similar test methods as those described herein, no lethal or sublethal effect was measured at a Cr sediment concentration of 2,600 µg/g (Bedard and Petro, 1997). Similarly, 10-day *Chironomus tentans* sediment tests were performed on several sediments collected from the same location in a 1991 study and no differences in survival or growth were found at Cr sediment concentrations ranging from 2,100 µg/g

TABLE 21. Spearman rank correlation analysis summary indicating significant negative (inverse) or positive (direct) correlation between biological endpoints and sediment physical and chemical parameters for Guelph stormwater pond 1997 samples.

Toxicity Endpoint	Mayfly Survival	Mayfly Growth	Midge Survival	Midge Growth	Minnow Survival
Bulk Concentration		+ Cadmium * - Oils & Greases *			- TOC * - Cadmium * - Copper * - Lead *
TOC corrected [a] Concentration		- Oils & Greases *			
%Fines corrected [b] Concentration		+ Zinc *			

[a] Chemicals that significantly covaried with sediment TOC included As, Cd, Cu, Pb, Zn and Oils and Greases.

[b] Chemicals that significantly covaried with sediment percent fines included Al, As, Fe, Hg, Mn and Zn.

* $p < 0.05$.

TABLE 22. Spearman rank correlation analysis summary indicating significant negative (inverse) or positive (direct) correlation between biological endpoints and sediment physical and chemical parameters for GTA stormwater pond 1997 samples.

Toxicity Endpoint	Mayfly Survival	Mayfly Growth	Midge Survival	Midge Growth	Minnow Survival
Bulk Concentration	- Oils & Greases * - Total PAH *	- TOC * - Oils & Greases **			
TOC corrected [a] Concentration	+ Cadmium *	+ Cadmium * + Mercury *		+ Cadmium *	
%Fines corrected [b] Concentration		- Chromium *			

[a] Chemicals that significantly covaried with sediment TOC included Cd, Cu, Hg, Pb and Zn.

[b] Chemicals that significantly covaried with sediment percent fines included As, Cr and Mn.

** $p < 0.05$; * $p<0.10$.

to 29,000 µg/g (US EPA, 1991). In comparison, the GTA SWP site had a Cr sediment concentration of 790 µg/g. In addition, the GTA sediment had a moderate amount of TOC which may have reduced the availability of the metal due to increased binding capacity. Another study also inferred that at sediment TOC levels exceeding 0.5%, any available Cr would likely exist in the less toxic trivalent form (Witt and Rodgers, 1991). This information implies the observed biological effects could not be related specifically to chromium.

The negative relationship between sediment TOC and mayfly growth potential is contrary to normal trends. Mayfly nymphs are burrowing animals that live below the sediment surface and rely on the surrounding sediment and detritus as food sources which they actively ingest (Zimmerman and Wissing, 1978). A limited supply of food will affect feeding activity and eventually cause subtle differences in growth performance. Sediment TOC below the optimal concentration of 2.0 mg/g can negatively affect mayfly growth (Bedard, 1989; Bedard and Petro, 1992). Interestingly, just the opposite was found for the GTA SWPs sediments. Sediments with relative low amounts of TOC supported good mayfly growth, while station T-3 with adequate amounts of TOC resulted in the poorest mayfly growth. It appears that sediment TOC concentration alone cannot adequately describe the test outcome for station T-3. It is possible that other sources of carbon were reflected in the elevated TOC measurement. This may have included carbon that was derived from the oils and greases measured in the sample.

Sediment oils and greases concentration appears to best explain the all or none pattern in biological effect. The highest oils and greases sediment concentration of 13.2 mg/g or 412 mg/g OC, for station T-3 coincided with the strongest negative biological effects. There was a slight toxic effect towards the mayfly (26% mortality) and high mayfly and midge growth reduction of more than 50%. All other GTA SWP sediments showed no adverse lethal or sublethal effects and had lower oils and greases sediment concentrations of under 3.4 mg/g or 200 mg/g OC.

A commonality between the Guelph and GTA sediment toxicity tests was the relationship between the concentration of oils and greases measured in the field sediments and degree of biological response. There was no corresponding sediment chemical analysis for oils and greases made on the samples prepared for toxicity testing e.g. sieved material. The number of samples that elicited a significant biological effect was relatively small and was inadequate in evaluating a dose-response relationship. The derivation of an effects-level concentration is useful for defining a chemical concentration that causes a specified level of impairment. Instead, a threshold concentration was used as an estimate of biological effect. The threshold concentration is that chemical sediment concentration above which effects were always observed for most test species.

In both studies, the two benthic invertebrates, in particular the mayfly *Hexagenia,* were more sensitive than the fathead minnow. Oils and greases sediment concentration normalized for differences in sediment total organic carbon was a more reliable indicator of toxicity as compared to the unnormalized sediment concentration. Overall, sediment oils and greases concentration of greater than 400 mg/g OC exhibited lethal and sublethal impacts, while oils and greases sediment concentrations above 200 mg/g OC were associated with marked sublethal growth effects. Oils and greases sediment concentration <200 mg/g OC had no effect either on organism survival or growth. These values should be viewed as guidelines since they were based on a relatively small sample size and may not necessarily apply to all SWP sediments. In addition, oils and greases encompasses a broad spectrum of petroleum-based substances and in itself is considered a gross

indicator of organic contamination. Unfortunately, the analysis of specific classes of organic compounds including PCBs and PAHs failed to provide any additional information with respect to identifying possible contaminant sources.

There were a number of characteristics that differed between the two studies. These site-specific features included the relative distribution and abundance of indigenous aquatic organisms in the test sediments, overlying water quality in the laboratory minnow bioassay, sediment organic matter quantity and sediment metal concentrations. All of these factors appeared to be related to the age of the stormwater pond. The median age of the ponds varied between the Guelph and GTA sites. The conditions found at the Guelph SWPs may be representative of those found in later stages of development. To facilitate intercomparisons between the Guelph and GTA sites, only those SWPs sharing the same land usage were examined. In this case, only SWPs classified as residential were taken into consideration. This examination is based on the assumption that none of the SWPs have been dredged.

For the most part, the Guelph SWPs, which had a median age of 13 years, had a corresponding average TOC that was three times higher than that measured at the younger GTA SWPs. In both study areas, Cd, Cu, Pb and Zn sediment concentrations were directly correlated to differences in sediment TOC. The more organic-enriched sediments found at the Guelph sites had consistently higher metal sediment concentrations. Therefore, as the ponds grow older the quantity of sediment deposited increases along with the TOC concentrations and associated trace metals. Several studies have also noted elevated concentrations for this same group of metals at Ontario SWPs (Licsko and Struger, 1995; Wren *et al.*, 1997). In this study, none of these metals were directly related to any adverse biological effect. In fact, metals such as Cd and Zn were correlated with improved benthic growth. The data suggest existing metal concentrations are not an immediate concern but do warrant continued monitoring given the age-dependent relationship. As the ponds grow older, these metals may eventually approach the PSQG-SEL concentrations and pose a greater toxicological concern. Efforts should also be made in determining to what extent these metal concentrations approximate those found under similar natural background conditions for each study location. This evaluation will better assess how the metal levels from stormwater runoff deviate from those due to atmospheric input or originate naturally due to sediment mineral composition.

Another variable that was unique to the Guelph SWP sediments was the relative number of extraneous oligochaetes found in most of the samples. Reynoldson *et al.*, (1994) demonstrated the ability of aquatic worms to negatively affect organism growth by as much as 90% for *Chironomus* and 50% for *Hexagenia*, depending on the worm density in the sample. The test sediments had either none, low, moderate or high numbers of oligochaetes, based on estimates made at the end of the midge test on sieved material caught on a fine-mesh screen. The differences in oligochaete density failed to parallel the differences measured in mayfly growth. The results were highly divergent and no obvious conclusion could be made. The presence of oligochaetes in the Guelph samples and their absence in the GTA samples could also be due to pond age and the degree of organic enrichment. At the same time, oligochaetes may be indicative of oxygen-poor conditions that occur *in-situ*. Other incidental evidence obtained from the fathead minnow bioassay support this idea. The Guelph sediments generated on average 3.5-times more un-ionized ammonia than the GTA test sediments. The main source of the ammonia seems to be directly from the sediment rather than through fish excretory functions. This was demonstrated by

the difference in ammonia measured between the two study areas, as well as the lack of ammonia generated in the negative control exposures. Although the un-ionized ammonia levels reached chronic concentrations, according to work by Thurston *et al.*, (1986) using similar size fathead minnows, it was not accompanied by obvious signs of chronic stress. Throughout the bioassay, the fish maintained normal contact with the sediment rather than avoiding the sediment and there were no changes in fish swimming behaviour. Stress was neither expressed in the form of weight loss based on estimated fresh weights made on pooled individuals. Average fish weight did not vary more than 5% among sites nor relative to the control. Other indicators may be required to properly measure chronic effects in fish. Together, the above observations suggest local water quality could become degraded through an increase in un-ionized ammonia from the sediment. Reduced oxygen and elevated ammonia could become limiting factors at certain sites and subsequently effect the ability of a site in sustaining a healthy aquatic community. It should be recognized that the laboratory tests do not reflect actual field conditions. To what extent these problems exist in the field could only be accurately assessed by routine monitoring of these variables.

5.0 CONCLUSIONS

Laboratory sediment toxicity tests conducted in 1997 on stormwater pond surficial sediments found few toxicological effects among sites. Among the Guelph sites, station G-8, resulted in a single adverse effect in the form of the smallest-sized *Hexagenia* nymphs. Multiple negative effects were detected for one of the GTA. Situated in a light industrial area, station T-3 elicited significant mayfly lethality of 25% and reductions in benthic growth of at least 50%. The majority of the stormwater pond sediments were considered to be nontoxic based on five toxicological endpoints.

Inorganic and organic chemical analyses on the sieved test material indicated few exceedences above the PSQG-Severe Effect Level concentration. This occurred only at one test site, station T-3, which had a chromium sediment concentration of 790 µg/g that surpassed the SEL by 7-fold. None of the chemical parameters were significantly correlated with the biological results. However, the concurrent field chemistry did provide the best overall predictor of relative differences in organism growth and to a limited degree with mayfly lethality. A threshold sediment concentration for oils and greases was estimated based on the range in observed biological effect. Oils and greases sediment concentration of 400 µg/g (corrected for organic carbon) was ordinarily associated with lethal effects using one of the test organisms (*Hexagenia limbata*) and 200 µg/g OC was associated with sublethal benthic growth impairment. Continued monitoring of oils and greases in stormwater pond sediments may serve as a useful predictor of potential biological effects.

The fathead minnow short-term bioaccumulation test examined chemical uptake of PCBs and pesticides from Guelph and GTA sediments. Whole-organism tissue concentrations were often comparable with concentrations found in the control fish, near trace concentrations and did not exceed federal or provincial tissue guidelines. The analysis was based on a single sample per treatment.

The interpretation of the toxicity results was limited due to the lack of an appropriate reference control sediment and only among site differences were examined. The inclusion of a

reference sediment would have provided useful information on naturally occurring contaminant levels for each study location. It is suspected that many of the trace metals and pesticide sediment concentrations encountered at the test sites are likely be similar to those found in nearby wetland areas. Possible exceptions could include copper and zinc sediment enrichment where concentrations were found to increase both with pond age and the nutrient content e.g. total organic carbon.

A unique characteristic that pertained only to the Guelph stormwater pond sediments was the presence of elevated ammonia concentrations generated in the fathead minnow bioassay and the relative abundance of indigenous oligochaetes found in most of the sediment samples. These variables may be indicative of degraded conditions in terms of anoxia and ammonia production. Sustained periods of low oxygen and elevated ammonia may influence the capacity of the pond to maintain viable fish populations or limit benthic productivity. Future field monitoring programs may want to consider these interactions, especially at older sites e.g. >10 years.

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